Quantifying the effects of overgrazing on mountainous watershed vegetation dynamics under a changing climate

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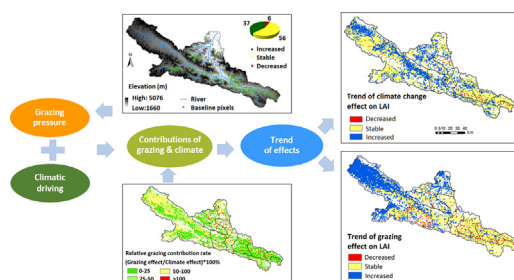[c] Center for Forest Disturbance Science, Southern Research Station, USDA Forest Service, Athens, GA 30602, USA
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HIGHLIGHTS

- Quantified pixel-scale grazing pressure dynamics using remote sensing-based leaf area index (LAI) data
- Overgrazing posed great challenges to shrub-grassland ecosystem recovery
- Separated grazing effects on LAI from inter-annual climatic variability
- Effects of climate on ecosystem services on rangelands may be underestimated without considering the long-term grazing

GRAPHICAL ABSTRACT

ABSTRACT

Grazing is a major ecosystem disturbance in arid regions that are increasingly threatened by climate change. Understanding the long-term impacts of grazing on rangeland vegetation dynamics in a complex terrain in mountainous regions is important for quantifying dry land ecosystem services for integrated watershed management and climate change adaptation. However, data on the detailed long-term spatial distribution of grazing activities are rare, which prevents trend detection and environmental impact assessments of grazing. This study quantified the impacts of grazing on vegetation dynamics for the period of 1983–2010 in the Upper Heihe River basin, a complex multiple-use watershed in northwestern China. We also examined the relative contributions of grazing and climate to vegetation change using a dynamic grazing pressure method. Spatial grazing patterns and temporal dynamics were mapped at a 1 km × 1 km pixel scale using satellite-derived leaf area index (LAI) data. We found that overgrazing was a dominant driver for LAI reduction in alpine grasslands and shrubs, especially for the periods of 1985–1991 and 1997–2004. Although the recent decade-long active grazing management contributed to the improvement of LAI and partially offset the negative effects of increased livestock, overgrazing has posed significant challenges to shrub-grassland ecosystem recovery in the eastern part of the study basin. We conclude that the positive effects of a warming and wetting climate on vegetation could be underestimated if the negative long-term grazing effects are not considered. Findings from the present case study show that assessing long-term climate change impacts on watersheds must include the influences of human activities. Our study provides important guidance for ecological restoration efforts in locating vulnerable areas and designing effective management practices in the study watershed. Such information is essential for natural resource

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management that aims at meeting multiple demands of watershed ecosystem services in arid and semiarid rangelands.

1. Introduction

Mountain grasslands provide direct and indirect benefits to people who depend on healthy and functioning ecosystems. In addition to the provision services of grasslands (food, fiber, and water), the importance of alpine grasslands for regulatory services (climate regulation, water regulation, soil conservation, carbon storage), cultural services (cultural diversity, spiritual and religious values, esthetic and recreational values), and supporting services (soil production, soil retention, oxygen production) is increasingly recognized (MEA, 2005; Bürgi et al., 2015; Lamarque et al., 2011). Although the ecosystem services (ES) provided by mountain grasslands have been demonstrated to be highly vulnerable to environmental and management changes in the past (Schröter et al., 2005), it remains unclear how they will be affected in the face of a combination of further land-use/cover changes and accelerated climate change (Schirpke et al., 2017).

Due to harsh climatic conditions, especially extreme temperatures and a shortened growing season, livestock production almost completely depends on natural forage produced in rangelands (SLN, 2014). However, intensive livestock production often has a negative effect on other ecosystem services, causing land degradation and affecting biogeochemical cycles (Steinfeld and Wassenaar, 2007). Overgrazing in alpine areas may result in soil and pasture degradation and the resultant decrease in their regenerative capacity, along with a reduction in vegetation production and biomass, lowered amination, nitrification (nitrogen fixation) and soil fertility (Steinfeld and Wassenaar, 2007). Therefore, rangeland grazing represents a common ecosystem disturbance in mountainous regions worldwide (Chen et al., 2015), and how this practice affects mountain grasslands ecosystem structure and functions and ecosystem sustainability under a changing climate has received tremendous attention in recent years (Sun et al., 2013a). For example, grassland overgrazing has been recognized as one of the key causes for the declines of grassland coverage and quality, loss of biological diversity, and land degradation from soil erosion in northwestern China (Chen et al., 2015; G. Sun et al., 2017a; L. Sun et al., 2017b). On the Qinghai-Tibetan Plateau, excessive livestock grazing is reported to have caused vegetation degradation and created barren soils over some 70,319 km^2 (Shang and Long, 2007). Therefore, understanding the effects of the driving forces behind grassland degradation has important ecosystem management implications in response to global change adaptation (Christensen et al., 2004; Chen et al., 2015; Gao, 2016).

Leaf area index (LAI) drives both the within- and below-canopy microclimate, determines and controls canopy water interception, radiation extinction, and water and carbon gas exchange and is thus a key component of biogeochemical cycles in ecosystems (Bréda, 2003). It is a critical parameter of ecosystem structure for modeling the processes occurring in the soil-plant-atmosphere continuum (Zarate-Valdez et al., 2012) and is essential for understanding vegetation growth and service functional response to climate change and land use (Daughtry et al., 1992; Musau et al., 2016). Changes in vegetation cover influence the hydrological cycle mainly through modulating the processes of canopy interception, evaporation and transpiration (hereafter referred to as evapotranspiration or ET), and infiltration (Liu et al., 2016; Sun et al., 2011a,b).

There are many factors that affect the vegetation LAI, and the dominant factors vary at different time and space scales (Yang et al., 2015). Our previous study (Liu et al., 2017; Hao et al., 2016a) showed that temporal and spatial LAI dynamics in the upper Heihe River basin of northwest China were affected by both climate variations and human

disturbances. Overgrazing was identified as one of the contributors to the decline in LAI in some areas and may have masked the positive climate warming effects in the study basin (Liu et al., 2017). Other factors, e.g., the soil type, also have a high effect on LAI distribution (Darvishzadeh and Skidmore, 2008). Previous results in this region showed that the soil type has relatively less impact on the annual or inter-annual variability in LAI compared with climate and overgrazing (Han et al., 1993).

Vegetation dynamics assessed by LAI are important in understanding the feedbacks between the biosphere and climate and the quantifying effects of human activities on ecosystem services such as water supply and carbon sequestration (Yang et al., 2015; Gao and Sang, 2017). For example, a decrease in LAI due to natural (e.g., droughts) or anthropologic (e.g., grazing) disturbances reduces ecosystem productivity and evapotranspiration (Hao et al., 2014; Han et al., 2015; Liu et al., 2016) but increases water yield (Sun et al., 2013). However, detecting grazing effects on plant communities is difficult because of spatiotemporal variability in vegetation structure and function as well as the lack of spatially explicit grazing data (Blanco et al., 2009; Wang et al., 2016); therefore, most of the existing studies were conducted at field scales for a couple of months (Bresloff et al., 2013; Su et al., 2010). In arid and semiarid regions, Blanco et al. (2009) thought that traditional field-based monitoring may not be appropriate to detect differences in grazing system effects on vegetation because the high temporal variability and spatial heterogeneity of this type of vegetation would mask the true impacts of grazing systems. The 'Gridded Livestock of the World' (GLW) dataset from the FAO's Animal Production and Health Division (Wint and Robinson, 2007) provides regional grazing distributions created through the spatial disaggregation of subnational statistical data (Han et al., 2014). However, the GLW dataset does not reflect the spatiotemporal variations of grazing densities and has a coarse spatial resolution of 0.05° (Wang et al., 2016). Long-term (decades) grazing changes and their impacts on LAI remain largely unknown at a regional scale (Liu et al., 2017; Hao et al., 2016a, 2016b). Very few comprehensive studies have evaluated grazing impacts on vegetation dynamics using multiple factors such as grazing location and duration and grazing density by vegetation types under a changing climate (Wang et al., 2016; Li et al., 2014). Advanced remote sensing techniques are available to quantify grazing effects by comparing the characteristics of vegetation coverage under grazing and nongrazing conditions (Wang et al., 2016; Archer, 2004). Blanco et al. (2009) detected the spatiotemporal patterns of vegetation using the Normalized Difference of Vegetation Index (NDVI) in a semiarid grazing ecosystem of Argentina. Li et al. (2012) examined human-induced vegetation changes and evaluated the impacts of land use policies in the Xilingol grassland region in Inner Mongolia, China, using the NDVI-based residual trend (RESTREND) method. Kawamura et al. (2005) and Li et al. (2014) investigated and identified the spatial distribution of grazing intensity based on the remote sensing NDVI in temperate grazing grassland system in Inner Mongolia. Wang et al. (2016) developed an approach to estimate regional grazing intensity and duration using NDVI data in a cold and arid grazing grassland system in the Qinghai-Tibetan Plateau. Washington-Allen et al. (2006) detected grazing impacts on vegetation in the United States using historical time series of remote sensing data. Yu et al. (2010) integrated the NDVI, aboveground biomass data, and theoretical livestock carrying capacity to estimate the grazing capacity. An increased incorporation of the current generation of remotely sensed data products into ES assessments can help drive a shift from reliance on simple spatial proxies of ESs to a more mechanistic focus on the ecological processes (Andrew et al.,

2014). However, few studies of the interactions between long-term spatiotemporal grazing patterns and associated vegetation dynamics at a regional scale are available (Feng and Zhao, 2011; Sha et al., 2014). In addition, previous grazing studies often focused on the numbers of livestock and thus evaluated grazing pressure using the number of livestock per grassland area (Li, 2011; Liu et al., 2010; Xin et al., 2011).

Detection of overgrazing or grazing management-related trends is considered challenging given the significant influences of inter-annual climatic variability in the rangelands. Separating the effects of grazing from climate also remains challenging due to the strong interactions among land management, climate, and ecosystem responses (Bastin et al., 2012; Pickup, 1989; Washington-Allen et al., 2006; Wessels et al., 2007). Previous studies have used minimally disturbed lands as reference areas to isolate grazing effects, although suitable benchmarks are usually difficult to identify (Bastin et al., 2012). Bastin et al. (2012) described a method to identify reference areas from multitemporal sequences of ground cover derived from remote sensing imagery.

The southern Qilian Mountains have remarkable vertical zonality and are the water source area of the Heihe River basin (Fig. 1a, b). Moreover, the southern Qilian Mountains have become an important forest and grassland nature reserve in western China (Wang et al., 2012). Animal husbandry represents the main economic pillar in this region, the output value of which accounts for 43% of local gross product (Wang and Li, 2008). Uncontrolled sheep and yak grazing on the Qilian Mountains has considerably increased over the recent past (Fig. 1c, d) and has inflicted adverse ecological effects on vegetation (Baranova et al., 2016). At present, the theoretical carrying capacity of grassland in the Qilian Mountain Protected Reserve is estimated as 853,800 SU (sheep units), and the existing livestock is 1.06 million SU, with an average overload of 40% (Wang et al., 2014). It is estimated that approximately

453,000 ha of shrub lands has been degraded to low-cover grassland (i.e., the grassland covered only approximately 5–20% with sparse grass and thus has poor utilization of pasture) at an altitude of 2300 m below the forest edge, which accounted for 65% of the total shrub land area in this region during recent decades (Guo et al., 2003). Therefore, overgrazing has become an important factor that affects the recovery of the forest-grassland ecosystem in the Qilian Mountain National Nature Reserve (Baranova et al., 2016).

The upper reach of the Heihe River (Fig. 1b) is located in the eastern part of the Qilian Mountains. In the mountainous upstream area, climate change has been the controlling factor both historically and at present (Cheng et al., 2014). The zones above the elevation of 3600 m are major water production areas of the Heihe River and account for 80% of the total runoff out of the Qilian Mountains (Wang et al., 2009). During the past three decades, runoff has increased due to the increase in summer precipitation and glacial melt due to a rise in air temperature (Yan et al., 2012; Cheng et al., 2014). The improved water conditions are due to global warming and increased precipitation, as well as ecological restoration in this region; both have resulted in a significant increase in vegetation coverage (Cheng et al., 2014). On the other hand, deforestation, overgrazing and grassland reclamation have caused serious degradation of the vegetation since the 1950s (Liu et al., 2017). However, the effects of grassland degradation on vegetation due to intensive overgrazing have not been well studied (Hao et al., 2016a). Understanding the vegetation dynamics of headwater watersheds under the coupled effects of climate and human activities is an integral part of a large ecohydrological research program on the Heihe River basin (Li et al., 2013).

This study focuses on the upper Heihe River basin as a case study. We developed an integrated method for mapping grazing areas using
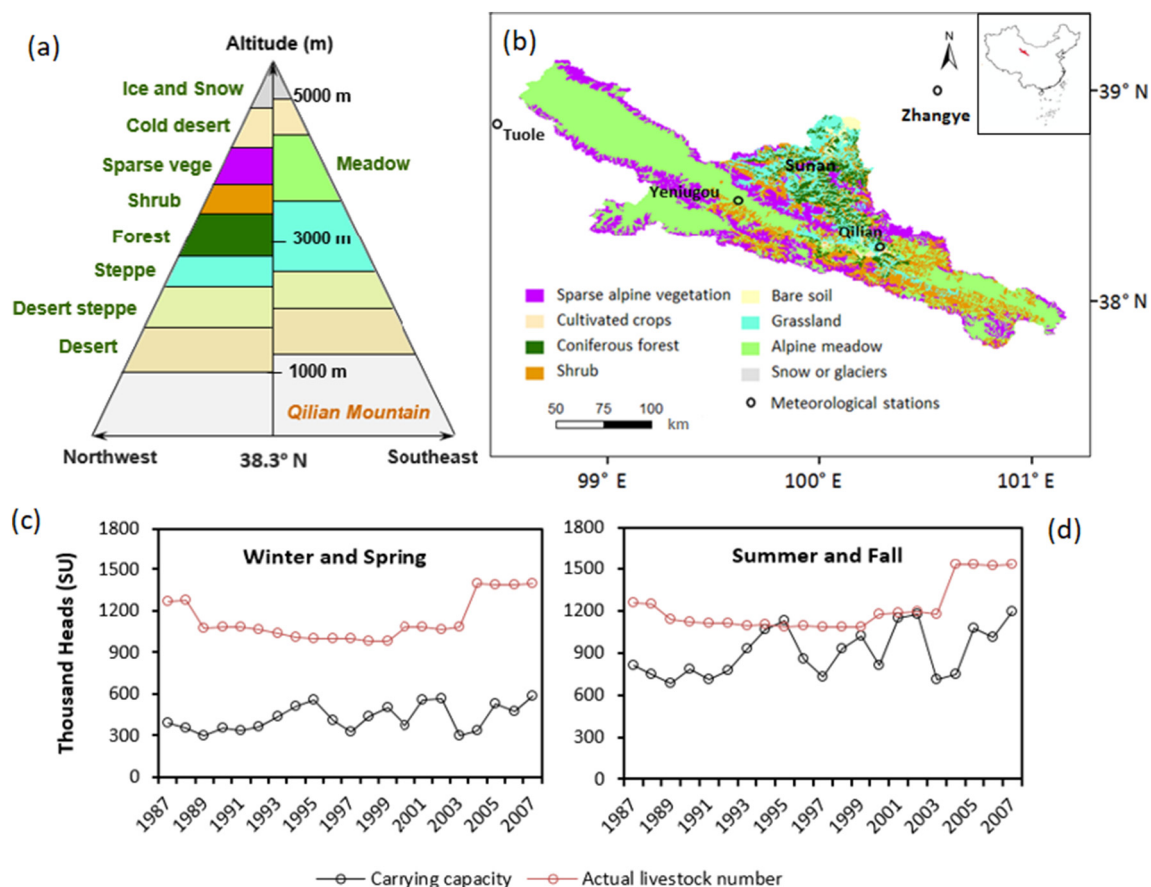
Fig. 1. Vertical (a) and lateral (b) vegetation distributions in the upper reach of the Heihe River Basin. The lines in the bottom graphs are carrying capacity and actual livestock in (c) the Winter-Spring Pastures (2400–3600 m, left) and (d) the Summer-Fall Pastures (3600–4100 m, right) in Qilian County (Sa, 2012).

high-resolution remote sensing techniques and linking historic grazing patterns with vegetation LAI changes and climate variability. We aimed to understand the long term (1983–2010) complex relationships between watershed vegetation changes and grazing disturbances under a changing climate. The specific objectives were to 1) detect the spatio-temporal grazing dynamics in the past decade by identifying the pixel-scale grazing pressure and duration using long-term remote sensing LAI data at a watershed level, 2) separate grazing effects on vegetation LAI under inter-annual climatic variability using a dynamic grazing pressure method (DGP), and 3) explore the implications of multiple drivers on vegetation LAI change and associated ecosystem function change in a multifunctional land-use watershed in an arid environment. The results contribute to the understanding of regional biophysical processes of global change and provide critical information for developing methods and strategies toward sustainable development in the study basin and beyond.

2. Materials and methods

2.1. Study basin

The upper reach of the Heihe River (Fig. 1) (37°41′–39°05′N, 98°34′–101°11′E) covers an area of approximately 10,005 km^2, with elevations ranging from 1700 m to approximately 5200 m. The long-term mean annual precipitation decreases from east to west and increases from north to south, with a high variability of 200–700 mm. Approximately 60% of the total annual precipitation falls in the warm growing season from June to September. The mean air temperature gradient is opposite to precipitation, with a high of approximately 6.2 °C at low elevations and a low of −9.6 °C at high elevations (Liu et al., 2016).

The major vegetation types in this basin include coniferous forest (*Picea crassifolia*), shrub, alpine grassland, alpine meadow (e.g., *Kobrecia parva*, *Kobresia humilis* and *Kobresia tibetica*), and sparse alpine vegetation (e.g., *Saussurea* DC., *Cremanthodium* DC. and *Rhodiola rosea* L.) (Fig. 1). The annual carrying capacity of grassland varies with different vegetation types, with the highest value (1.33 SU ha^{-1}) in shrub-grassland and the lowest (0.73 SU ha^{-1}) in alpine grassland (Table 1). Grassland degradation is common due to historically intensive overgrazing. Overgrazing is more widespread in the Winter-Spring Pastures at elevations of 2400–3600 m than in the Summer-Fall Pastures at elevations of 3600–4100 m, with the former being the more convenient and having the longer grazing period (Fig. 1).

2.2. LAI, climate, and grazing datasets

We used long-term time-series (1983–2010) satellite-derived Global Land Surface Satellite (GLASS) LAI datasets to identify the grazing effects during the past three decades. The GLASS LAI product (version 3.0) (BNU, 2016) has an integerized sinusoidal (ISIN) projection with a temporal resolution of 8 days. During the period from 1983 to 2000, the spatial resolution is 0.05° (approximately 5 km at the equator) generated from LTDR AVHRR reflectance data with a geographic latitude/longitude projection. From 2000 to 2010, the dataset is derived from MODIS surface-reflectance data

Table 1
Livestock carrying capacity and yield for key grasslands near the Qilian Mountain area.[a]

Grassland Types	Annual livestock carrying capacity (SU ha^{-1})	Annual mean yield of fresh grass (kg ha^{-1})
Alpine grassland	0.73	1388
Alpine meadow	1.27	2224
Shrub-grassland	1.33	2164
Sparse forest grassland	1.08	1761

[a] Data from Yu (1983). The livestock number is equivalent to sheep units (SU) where one horse is 6 SU, one head of cattle is 5 SU, and one goat is 0.8 SU, according to Zhang and Liu (1992).

with a sinusoidal projection at a spatial resolution of 1 km. Extensive validation of LAI accuracy has been performed to ensure that the temporally and spatially continuous fields of LAI are reasonable during 1983–2010 (Xiao et al., 2014, 2016).

The monthly meteorological observation dataset (1983–2013) (China Meteorological Data Service Center) was used to construct a drought index (DI) defined as the ratio of precipitation (P) and potential evapotranspiration (PET) (DI = P/PET). This DI is a variant of the dryness index originally defined by Budyko (1974). PET was estimated using the Hamon method (Hamon, 1963) as a function of saturation vapor pressure (a function of air temperature) weighted by daytime length and inversely proportional to air temperature.

The annual livestock data in Qilian County were obtained from the Census of Haibei (1990–2010), and township livestock data were obtained from the Census of Qilian County (1995–2003) and Sunan County (1985–2010). Based on these datasets, the annual livestock data for the period of 2000–2010 were interpolated and scaled to each township. The total livestock number is equivalent to sheep units (SU), where one horse is 6 SU, one head of cattle is 5 SU, and one go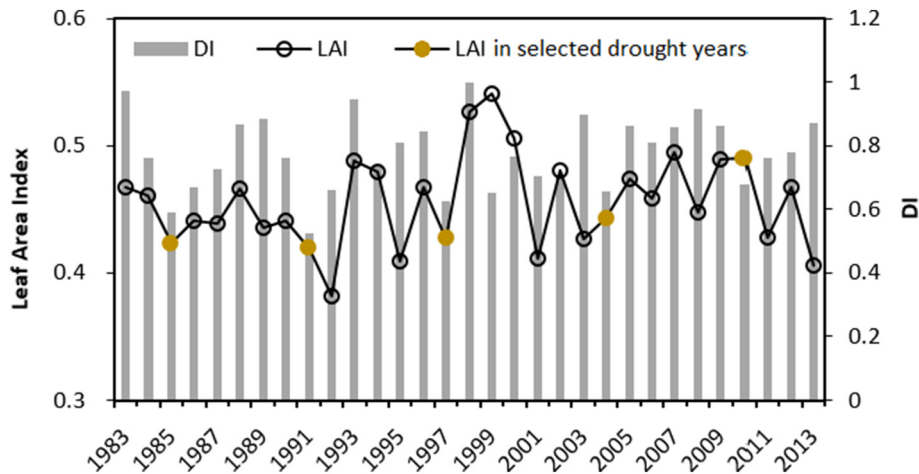at is 0.8 SU, according to Zhang and Liu (1992). The vegetation type maps used in this study had a spatial scale of 1:100,000 (Zhang et al., 2016).

2.3. Determining baseline pixels with minimum disturbance

We modified the dynamic reference-cover method (Bastin et al., 2012) to determine baseline pixels (i.e., reference pixels) that represent areas with minimal grazing disturbance for the long period of 1983–2010. We assumed that a ground cover that endures dry periods indicates a benchmark that defines a resilient and productive rangeland landscape, including low erosion potential and high landscape functionality (Ludwig et al., 1997). It is easier to identify the grazing effects that have the most impacts on vegetation LAI under drought conditions than during normal years. The most resilient and productive cover is likely to be least affected by droughts and grazing disturbances. Moreover, we assumed that there is a large difference in LAI between each regular pixel and the baseline pixel during dry periods.

The rules above allowed reliable and robust identification of baseline pixels subject to grazing. We first calculated pixel-level mean LAI during the dry months (October throughout May) during 1983–2010 and then selected the pixels with the minimum LAI by land cover type. Then, we selected pixels with relatively high LAI values (top 10%) to represent those pixels most resilient to drought and disturbances for each vegetation cover type subject to grazing (See S1). Three grazing vegetation types (i.e., alpine meadow, alpine grassland, and shrub) were used to separately determine the baseline LAI pixels.

2.4. Quantifying long-term grazing effects on LAI at a regional scale

We modified the method proposed by Bastin et al. (2012) to quantify the general pattern of long-term grazing effects on LAI for the entire basin. We assumed that LAI changes mainly depended on climate change and grazing change, which included both negative overgrazing effects and positive grazing management (e.g., grazing exclusion and rotation). Based on this assumption, long-term grazing effects at a regional scale during 1983–2010 were quantified by the following method.

First, we determined drought years by comparing the Drought Index (DI) for rainy seasons (June–September) (Fig. 2). For each suitable drought year t, the pixel-level actual and baseline LAIs were spatially averaged to define a regional value; then, the difference between regional actual and baseline LAI was calculated as:

$$dLAI(t) = LAI(t) - LAI(t)_{BAS}.$$

Fig. 2. Leaf area index (LAI) and growing season DI (P/PET) from June to September during 1983–2013 (the brown spots are selected drought years).

Change in the *dLAI* between successive drought years of t_1 and t_2 (Fig. 2) is denoted as $\Delta dLAI$ and is calculated as follows:

$$\Delta dLAI(t_1, t_2) = dLAI(t_2) - dLAI(t_1).$$

dLAI represents the seasonal LAI for the region in a particular drought year and $\Delta dLAI$ represents an aggravating or reduced grazing effect compared with the previous drought year. $\Delta dLAI$ thus provides a measure for separating the effects of grazing management from those of climate variability. We denoted ΔLAI_G as the grazing effects alone in contrast to the climatic effects on LAI at the pixel level and different temporal scales. More details can be found in Bastin et al. (2012).

2.5. Identification of spatiotemporal grazing dynamics at a pixel scale

We used the methods proposed by Wang et al. (2016) to identify the spatiotemporal grazing dynamics during 2001–2010 at a pixel scale across the entire watershed. The first step was to examine whether the pixel was under grazing from time i to j using two indices, G_{LAI} and $G_{\Delta LAI}$ (Wang et al., 2016). The second step was to calculate the grazing duration and grazing pressure.

We defined G_{LAI} as the relative difference between LAI values for grazed and ungrazed pixels. We defined $G_{\Delta LAI}$ to represent relative differences in LAI changes over time between two pixels under grazing and nongrazing conditions.

$$G_{LAI} = \frac{LAI}{LAI_N}$$

$$G_{\Delta LAI} = \frac{\Delta LAI}{\Delta LAI_N} = \frac{LAI_j - LAI_i}{LAI_{Nj} - LAI_{Ni}} \qquad (j > i \geq 1)$$

where LAI_N is LAI for pixels under nongrazing conditions; LAI_i and LAI_j are pixel-level LAI at times i and j, respectively, both with an eight-day interval; and LAI_{Ni} and LAI_{Nj} represent nongrazing pixel-level LAI at times i and j, respectively. The nearest baseline pixels with the same vegetation type as the regular pixels were used as reference nongrazing pixels in calculating $G_{\Delta LAI}$.

We assumed that the annual LAI time-series for undisturbed pixels follow a parabola function, i.e., the LAI values increase and reach a maximum value and then decrease for the remaining period. We also assumed that the increase in LAI in the grazing pixel was less slow than that of the nongrazing pixel before the LAI reaches its peak but fell faster after the peak. Based on these assumptions, a pixel was identified as being grazed if $G_{LAI} < 1$ and $G_{\Delta LAI} < 1$ prior to the period when LAI reaches

the maximum. For the remaining period after LAI reaches the peak, the pixel must meet the conditions of $G_{LAI} < 1$ and $G_{\Delta LAI} > 1$ to be classified as grazed.

We defined grazing duration (*GD*, days) and grazing pressure (*GP*, SU ha^{-1}) for a pixel to identify the spatiotemporal grazing dynamics. The grazing duration for a pixel was then calculated from the time interval between two adjacent 8-day remote sensing images. *GP* is defined as the carrying livestock number per grassland area in a certain period and was classified into four levels in this study basin: light grazing (0–2.4 SU ha^{-1}), moderate grazing (2.4–4.0 SU ha^{-1}), overgrazing (4.0–5.6 SU ha^{-1}), and severe grazing (>5.6 SU ha^{-1}) (Li, 2011). The Grazing Pressure (*GP*) during a certain period was calculated as:

$$GP = \frac{|dLAI_G - dLAI_N|}{\sum |dLAI_G - dLAI_N|} \times LS$$

where

$$dLAI_G = LAI_{Gj} - LAI_{Gi}.$$

LAI_{Gi} and LAI_{Gj} are grazing pixel LAI at times i and j, respectively, and LS (SU ha^{-1}) is annual livestock numbers in each township.

2.6. Contributions of grazing and climatic effects on LAI changes at the pixel scale

Change in LAI (ΔLAI) during 2001–2010 at the pixel scale was assumed to have been caused by grazing (ΔLAI_G) and climate change (ΔLAI_C) during 2001–2010:

$$\Delta LAI = \Delta LAI_C + \Delta LAI_G$$

where

$$\Delta LAI_G = \Delta(dLAI_G - dLAI_N)$$

ΔLAI_G means the pixel LAI changes due to the grazing effect from time i to j. $\Delta LAI_G < 0$ means that the grazing change leads to a decreasing LAI. ΔLAI_C represents the LAI changes due to climate effects from times i to j, calculated as:

$$\Delta LAI_C = \Delta LAI - \Delta LAI_G.$$

The grazing contribution rate (*RG*) during a certain period was calculated as:

$$RG = \frac{\Delta LAI_G}{\Delta LAI} \times 100\% (\Delta LAI \neq 0)$$

Several scenarios can occur for *RG* that represent different contributions from grazing effects:

1) When LAI at one pixel increases ($\Delta LAI > 0$) and grazing leads to a reduction in LAI ($\Delta LAI_G < 0$), *RG* becomes negative, which indicates a negative contribution to LAI from grazing.
2) When LAI for a pixel increases ($\Delta LAI > 0$) and $\Delta LAI_G > 0$, *RG* becomes positive, which indicates a positive contribution of grazing.
3) When LAI for a pixel decreases ($\Delta LAI < 0$) and $\Delta LAI_G < 0$, *RG* becomes positive; we still define this a negative contribution of grazing.
4) When LAI for a pixel decreases ($\Delta LAI < 0$) and $\Delta LAI_G > 0$, *RG* becomes negative; we still define this as a positive contribution of grazing.
5) When the change in LAI is minor, i.e., $\Delta LAI = 0$, the contributions from grazing are assumed to be 50%, either positive if $\Delta LAI_G > 0$ or negative if $\Delta LAI_G < 0$.
6) When $\Delta LAI_G = 0$, *RG* is zero, which indicates no contribution of grazing.

Therefore, we can essentially use the absolute value of *RG* to evaluate the relative contributions of grazing effects on LAI.

Then, the climate contribution rate (*RC*) during a certain period was calculated as:

$$RC = 1 - RG$$

The relative grazing contribution rate (*R_{GC}*) during a certain period was calculated as:

$$R_{GC} = \left(\frac{|RG|}{|RC|}\right) \times 100\%$$

R_{GC} is the ratio of the absolute value of *RG* to the absolute value of *RC*. $R_{GC} < 100$ means that the grazing effect is less than the climate effect, otherwise the opposite.

The nonparametric MK test (Mann, 1945; Kendall, 1975) was applied to analyze the LAI and grazing parameter trends. We identified three trend types, 'Increased', 'Stable', and 'Decreased', for the LAI and grazing parameters according to their standard test statistic *Z* values in the MK test.

3. Results

3.1. LAIs for undisturbed reference locations

The Baseline LAI Pixels are mostly located in the middle and eastern basin with higher vegetation LAI and are concentrated in the perimeters of rivers and the feet of mountains (Fig. 3). The meadow pixels account for 38% of all Baseline LAI Pixels, 37% of shrubs, and 16% of grasslands. The 'Stable' vegetation LAI pixels account for 56% of all Baseline LAI Pixels, and the 'Increased' and 'Decreased' pixels account for 37% and 6%, respectively.

3.2. Long-term grazing effects on LAI at a regional scale

In recent decades, both overgrazing and active grazing management contributed to LAI variations in this basin. Overgrazing was the dominant negative driver for the reduction in vegetation LAI. The most affected vegetation types are shrubs and alpine grasslands, especially during the periods of 1985–1991 and 1997–2004 (Fig. 4). With the implementation of grazing exclusion and grazing rotation policy in grassland, the grazing management contributed to the improvement of LAI, which partially offset the negative effects due to increasing livestock.

3.3. Grazing patterns in the upper Heihe River basin

The annual mean grazing duration during 2001–2010 varied from the maximum grazing days (~200 d) in western areas to fewer days in the northern and eastern areas of this basin (Fig. 5a). Light grazing pressure (0.0–2.4 SU ha^{-1}) was found in western and northern areas, whereas moderate grazing or overgrazing pressure (>2.4 SU ha^{-1}) was found in the eastern area of this basin (Fig. 5b). The longest grazing duration occurred in spring and summer, which dominated the annual spatial pattern of grazing duration, followed by autumn and winter. Autumn had a higher grazing pressure compared with other seasons.

Alpine meadow in the western Yeniugou area had maximum grazing days with low grazing pressure due to nomadic and scattered yak grazing in all years (Fig. 6a, b). Alpine grassland in the northern Sunan region had less grazing days as well as lower grazing pressure (Fig. 6c, d). Shrub had less grazing days in the northern and eastern areas;
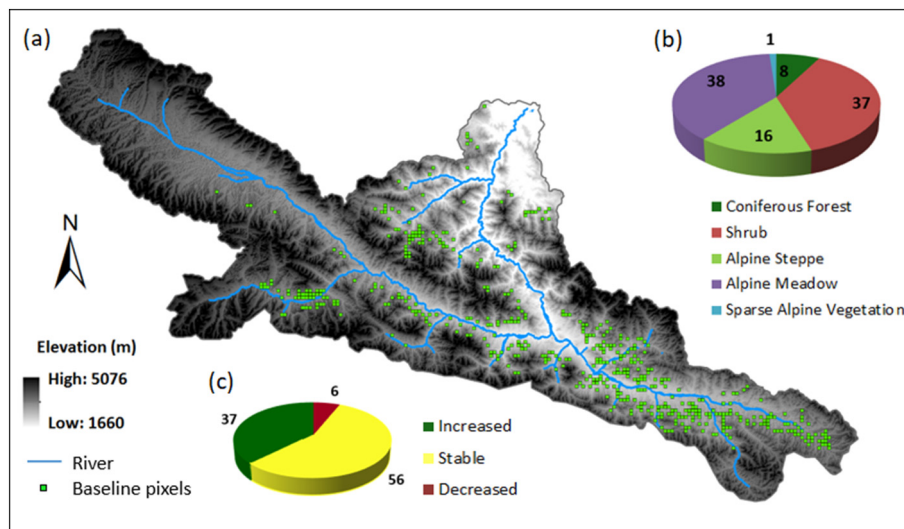
Fig. 3. Spatial distribution of baseline pixels (a), composition of land cover types (b), and temporal trends of Leaf Area Index (LAI) for baseline pixels in terms of proportions (%) of areas (c) during 2001–2010.
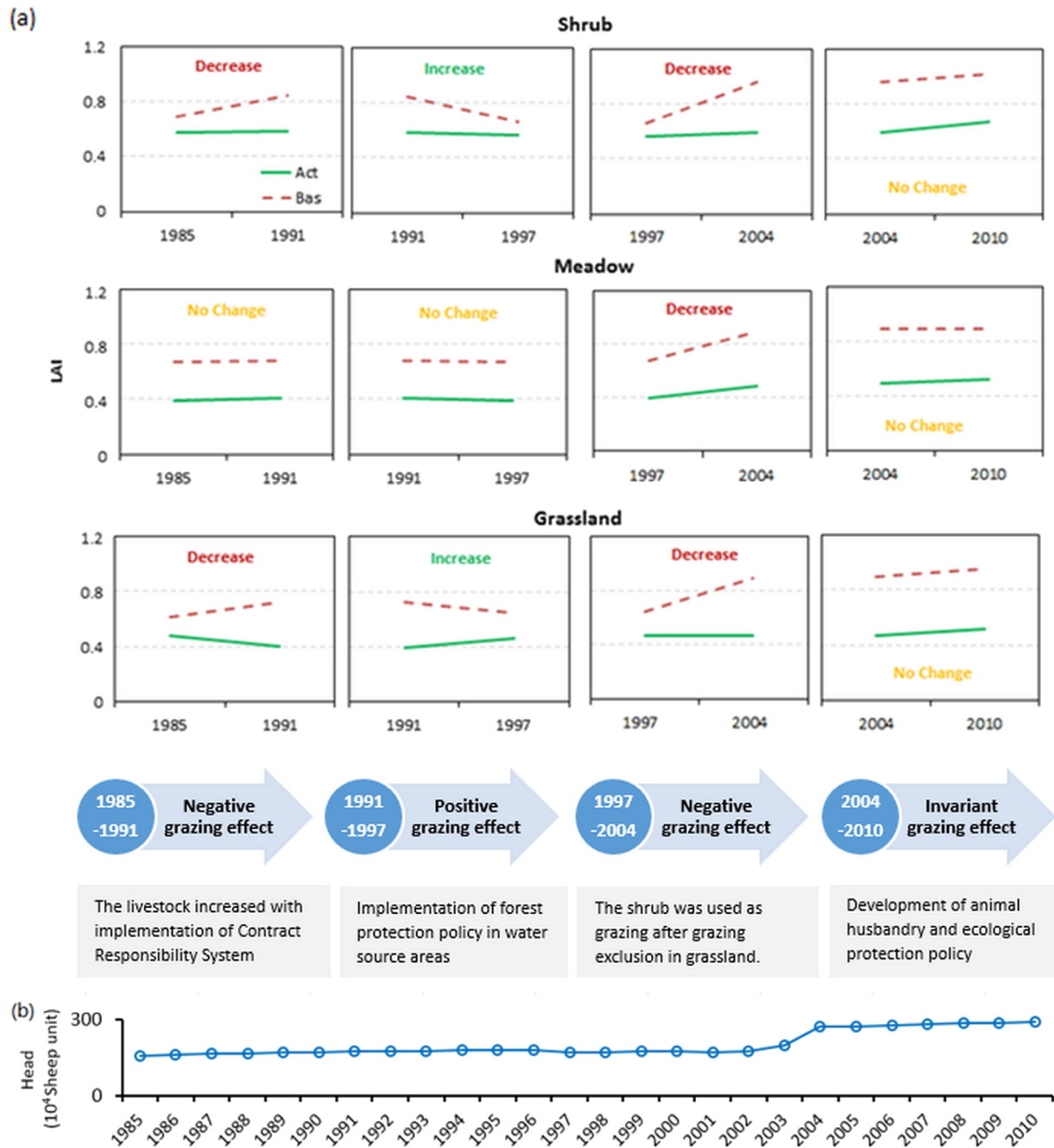
Fig. 4. (a) Changes in averaged actual (green) and baseline (red) levels of LAI for shrub, alpine meadow, and alpine grassland after allowing for seasonal effects between the two dry years of 1985 and 1991, 1991 and 1997, 1997 and 2004, and 2004 and 2010, respectively. The distance between the solid and dashed lines (*dLAI*) represents the reduction in LAI due to grazing, and the slope change of the two lines (*ΔdLAI*) represents an aggravating or reduced grazing effect compared with the previous year (i.e., a slope of the dashed line greater than that of the solid line represents aggravating grazing, and the opposite condition represents reduced grazing or positive grazing management. The same slope indicates no change with grazing for the two periods.) The graph (b) at the bottom shows the annual livestock heads in this region during 1985–2010.

however, it carried the most livestock numbers and therefore suffered maximum grazing pressures in the eastern area (Fig. 6e, f).

3.4. Contributions of grazing and climate change to LAI dynamics

The annual LAI trends for the three grazing vegetation types of alpine meadow, alpine grassland, and shrub were examined; the areas with annual LAI classified as 'Stable' accounted for approximately 57% of the total study area, and the rest were classified as 'Increased' (40%) and 'Decreased' (3%) (Fig. 7a). Compared with the other two types of

vegetation cover change, the decreased areas were minor and distributed in the eastern area.

For the areas with 'Increased' LAI, negative grazing contributed the most (≤−25%) in the western part of the basin, which means that the grazing leads to the most reduction in vegetation LAI in this area (Fig. 7b). The positive grazing contribution is distributed in the northern and eastern parts of the basin, i.e., the grazing leads to an increase in vegetation LAI in these areas (Fig. 7b). For the 'Stable' area, the negative grazing is distributed in most of the western and northern parts of the basin (Fig. 7c). For the 'Decreased' area, the negative grazing is scattered throughout the entire basin and the most negative grazing effect (≥50%)
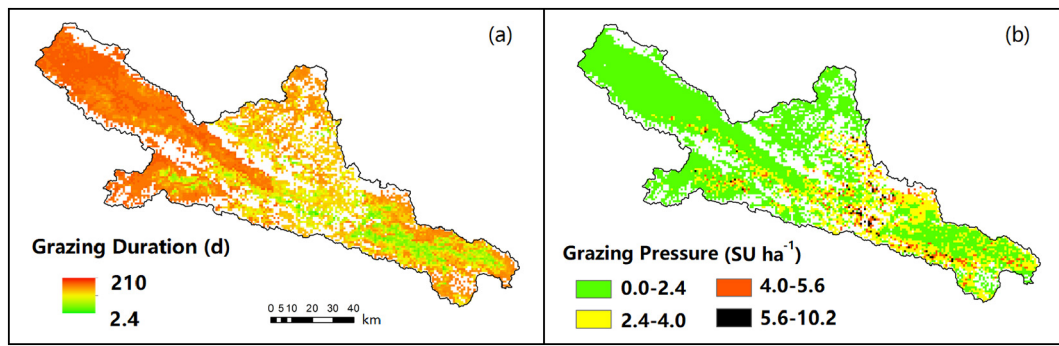
Fig. 5. Spatial patterns of annual (a) grazing duration (GD, days) and (b) grazing pressure (GP, SU ha^{-1}) during 2001–2010.

is distributed in most of the northern and eastern parts of the basin (Fig. 7d).

The contributions of climate overwhelmed that of grazing in most areas of this basin (Fig. 8a), with the higher grazing contribution (R$_{GC}$ > 50) in western and northern areas of the basin. The increasing trend of climate contribution (Fig. 8b) is consistent with the increasing trend in vegetation LAI pattern (Fig. 7a), which suggests that the vegetation LAI change pattern during this period was dominated by climate change. However, the decreased vegetation LAI trend pattern (Fig. 7a) is

the result of the combined effects of the negative climate effect trend (Fig. 8b) and the increasing grazing effect (Fig. 8c), which indicates that the grazing may have aggravated the negative effects of climate and led to a decreased LAI in the eastern region (Fig. 7a, d) and partially offset the increased LAI impacts of warming and wetting climates (Hao et al., 2016a, 2016b) in the western and central areas (Fig. 7a). The contributions to the observed annual mean change in LAI during 2001–2010 suggest that the general positive effects of climate overwhelmed the negative effects of grazing over the entire basin
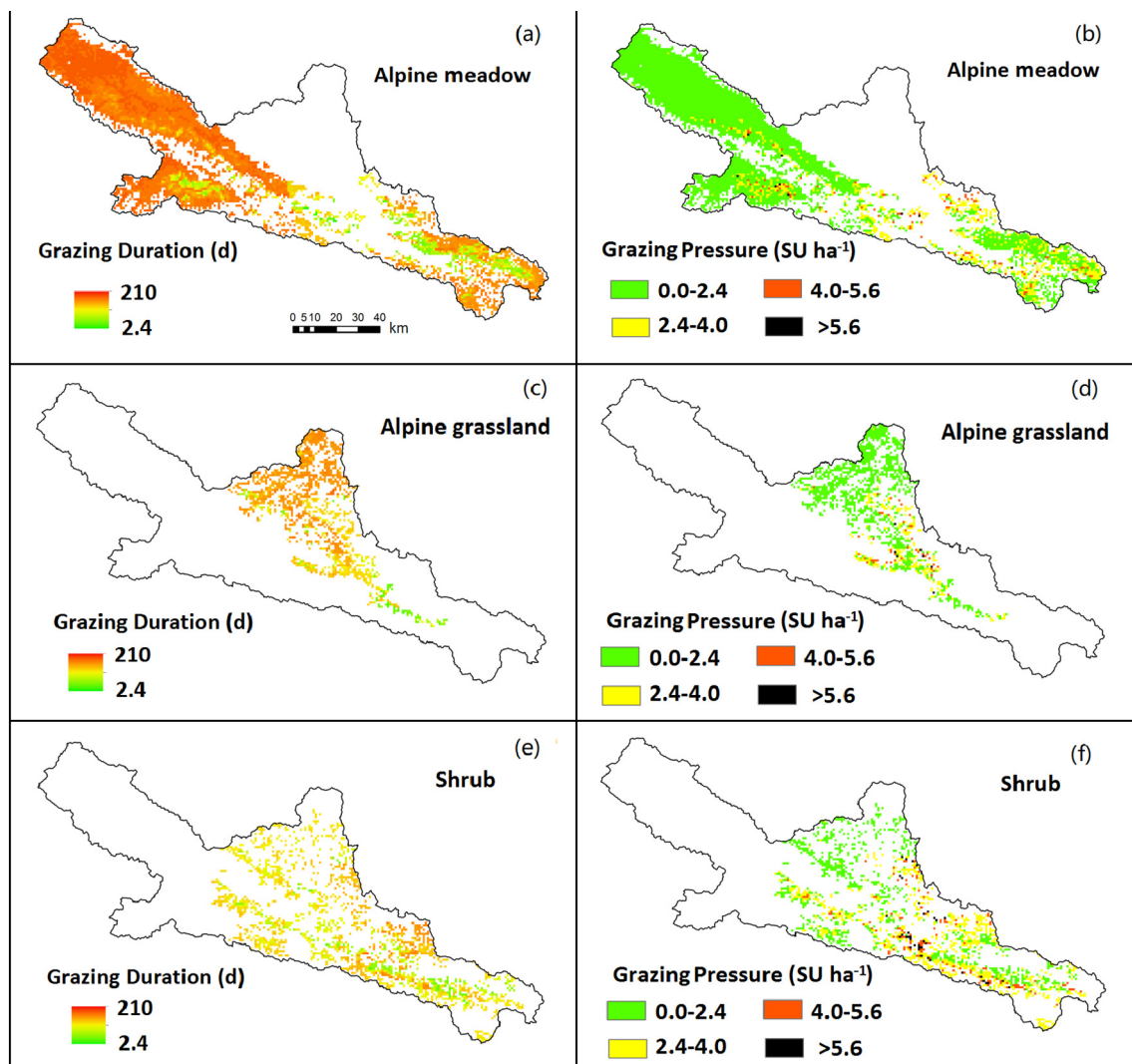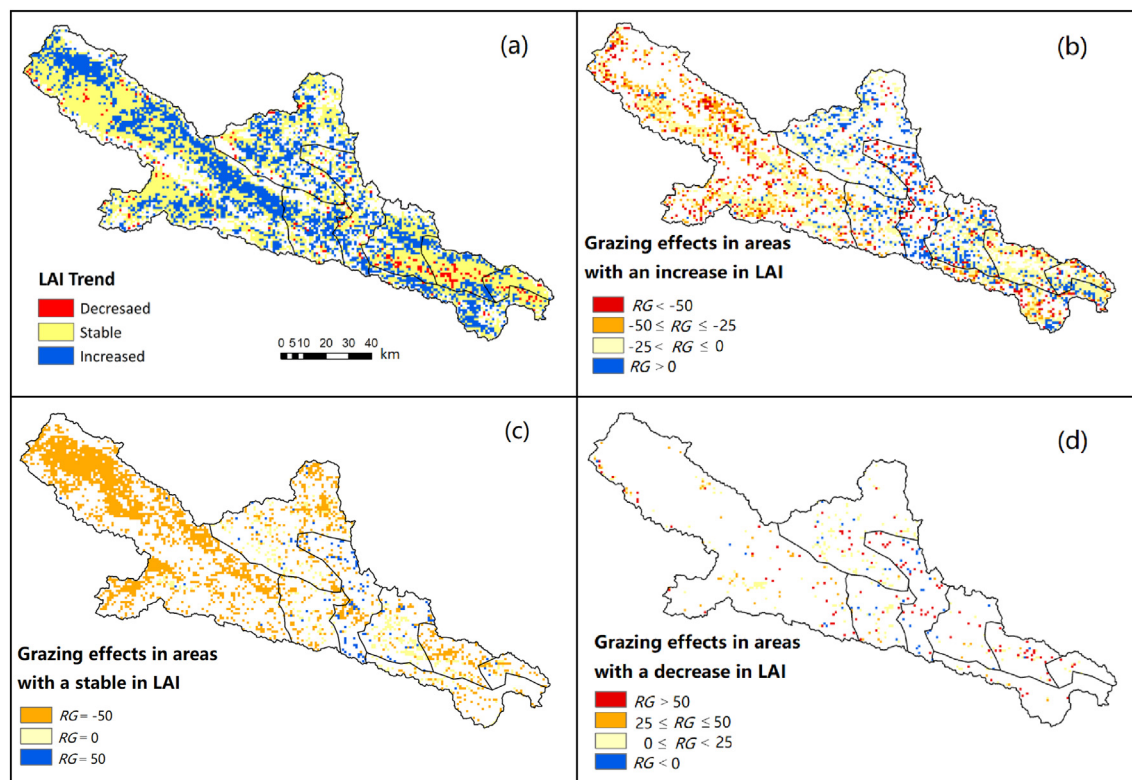

Fig. 6. Spatial patterns of annual mean grazing duration (GD, days), and grazing pressure (GP, SU ha^{-1}) (2001−2010) with three grazing vegetation types of (a) and (b) alpine meadow, (c) and (d) alpine grassland, and (e) and (f) shrub, respectively.

Fig. 7. Grazing effects (RG, %) on the trend of annual vegetation leaf area index (LAI) for the Upper Heihe River Basin during 2001–2010. (a) Annual vegetation LAI trend, (b) grazing contribution (RG, %) in areas with an increase in LAI (negative RG indicates negative contribution of grazing), (c) no change in LAI (positive ($RG = 50$%) indicates contribution for an increase in LAI and negative ($RG = -50$%) indicates decrease in LAI), and d) grazing contribution (RG, %) in areas with a decrease in LAI (positive RG indicates negative contribution of grazing).
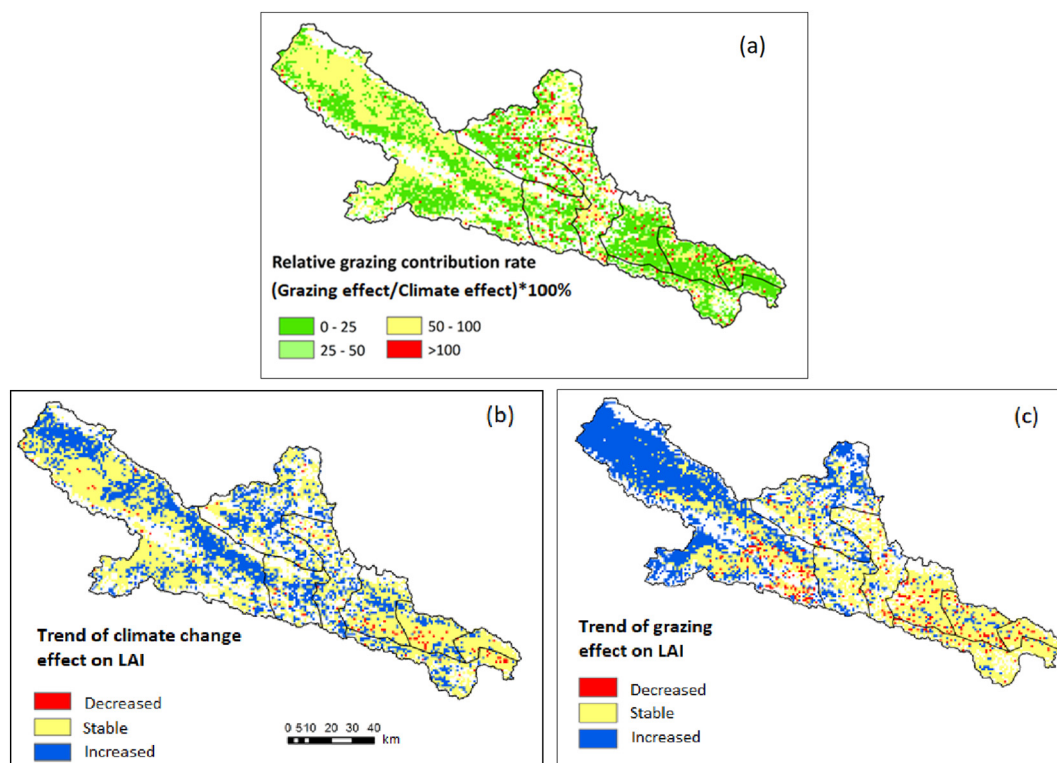
Fig. 8. Relative grazing contribution rate (R_{GC}) (a) and trends of effects of (b) climate change (RC) and (c) negative grazing (RG) over time on leaf area index (LAI) for the Upper Heihe River basin during 2001–2010. Three distinct areas in the trends of effect are identified as Decreased, Stable, and Increased. $R_{GC} < 100$ means that the grazing effect is less than the climate effect, otherwise the opposite.

(Fig. 8, Fig. 7a); however, the positive effects of a warming and wetting climate on vegetation could be underestimated if the negative long-term grazing effects are not considered.

4. Discussion

4.1. Relationships of grazing and vegetation ecosystem services in the mountainous basin

Mountainous rangeland in the Heihe River basin provides important ecosystem services by means of provisioning (forage for livestock, freshwater), support (soil formation and conservation, nutrient cycling), regulation of water and climate, and culture (e.g., cultural identity and diversity, tourism) (Li et al., 2013). This means that rangelands have multiple, and sometimes conflicting, values for stakeholders (Díaz et al., 2006). In the past decades, many of the inland river basins in China's northwestern arid region have experienced a common challenge of water shortages. In the Heihe River basin, the consumption of water dramatically increased with the booming population and rapid economic development (Li et al., 2013). At present, the most important issue in this basin is the contradiction between the growing demand of socio-economic development and the increasing water shortages.

The upper Heihe basin generates nearly 70% of the total river runoff, which supplies agricultural irrigation and greatly benefits socio-economic development in the middle and lower basin (Chen et al., 2005). Vegetation pattern is a key factor that affects the water balance and catchment water yield (Yang et al., 2015). However, with global climate change and population growth in the past few decades, ecosystem degradation in this basin has occurred and the key ecosystem components have changed, which has resulted in reductions in the individual and multiple ecosystem services that support people's livelihoods, health, and way of life (Li et al., 2013). In the midstream area of the Heihe River basin, natural oases have been gradually replaced by artificial oases (Chen et al., 2005). Natural oases in the downstream area have experienced some desertification due to reduced water from the middle reaches (Li et al., 2013). The discharge in the lower basin of the river has significantly decreased, and >30 tributaries as well as the terminal lakes have dried up (Chen et al., 2005).

Alpine vegetation ecosystems can regulate radiation and energy budgets, which results in changes in the hydrological processes in frozen soil and increased mountain runoff (Ji et al., 2007). Sparse alpine vegetation and alpine meadow dominates the high-altitude regions that contribute most to the water yield and river runoff (Yang et al., 2015). However, in the upstream area of the Heihe River basin, deforestation, overgrazing and grassland reclamation have caused serious degradation of the vegetation since the 1950s (Li et al., 2013). The carrying capacity of the rangeland has been substantially degraded (Chen et al., 2005).

Fig. 4 shows the livestock number change and socio-economic management policies during different periods in this basin. By 2010, the number of livestock (Sheep units) had grown from 1500 thousand in 1980s to approximately 2900 thousand (Fig. 4). In the 1970s, the government called the pastures "take grain as the key link". A large area of natural grassland and forestland was cultivated as arable land. Since the 1980s, the government has encouraged the development of animal husbandry. Livestock increased with the implementation of the Contract Responsibility System but the boundaries of grazing were unclear. During the 1990s, the forest protection policy in water source areas was implemented. The grassland paid transfer system was implemented in the pastoral areas, and households were compensated for grassland use. Local animal husbandry has entered an accelerated development period. To protect grassland ecology, the Chinese government has invested 2.35 billion RMB since 2000 to implement ecological restoration policies in the Heihe River basin. Grazing exclusion and afforestation were implemented, and intensive grassland animal husbandry

was vigorously developed. However, although a series of ecological restoration policies and ecological protection projects have been implemented in recent decades in this region, disorder by overgrazing without permission remains very common and thus contributes to vegetation deterioration. It can be seen that the government management policy was the main driving force of livestock numbers during the past decades. Before the year 2000, the main purpose of the government's management policy was to promote economic development; after 2000, the government began to pay more attention to ecological environment protection. This study indicated that the current grazing pressure should be alleviated and that grazing management (e.g., grazing exclusion and grazing rotation) needs to be further strengthened in this region.

4.2. Effects of grazing on vegetation LAI variations

Before 2000, overgrazing was the dominant negative driver for the reduction in vegetation LAI; the most affected vegetation types were alpine grassland and shrubs, especially for the periods of 1985–1991 and 1997–2004 (Fig. 4). During the recent decade, watershed-wide mean LAI has had an increasing trend over time for all vegetation types (Liu et al., 2017), and the recovery of vegetation has been documented by remote sensing and ground studies (Deng et al., 2013; Yan et al., 2016). Our previous study concluded that both climate change and grazing were responsible for the detected long-term LAI trend during the past decade (Hao et al., 2016a; Liu et al., 2017). However, remote detection of overgrazing or grazing management-related trends in the presence of inter-annual climatic variability in the rangelands has not been examined. This study separated the grazing effects on LAI from inter-annual climatic variability and found that heavy grazing pressure still existed in the eastern basin (Fig. 5b), which has significantly decreased LAI as well as the highest mean annual LAI value in this basin (Hao et al., 2016a, Liu et al., 2017).

The grazing contribution markedly increased in the western and northern parts of the basin from 2001 to 2010 (Fig. 8c). This indicates that the negative grazing contributions to vegetation LAI (reducing LAI) have partially offset the positive influence of climate (increasing LAI). Without considering the long-term grazing change, the contribution of climate could have been underestimated in this study.

The positive contribution to vegetation LAI (e.g., the blue points in Fig. 7b) results from calculation error and grazing pressure. The existing field observations in alpine grassland found that when water and heat are suitable, light or moderate grazing pressure is more beneficial to increase LAI than not grazing, i.e., the LAI in areas with light or moderate grazing is higher than the LAI in nongrazing areas (Su et al., 2010). The positive contribution was mostly distributed in northern Sunan and its eastern area with light or moderate grazing pressure (Figs. 5, 6) and a more favorable climate (Fig. 8b). Therefore, the grazing contribution was positive. In addition, the vegetation type map used in this study was from 2000 (Zhang et al., 2016). However, the massive afforestation campaigns in the north of the basin have occurred since 2000 and could also impact the positive contribution results (Hao et al., 2016a, 2016b).

Our analysis showed that grazing has intensified the negative effects of warming and drying climate on vegetation LAI in the eastern region, whereas grazing has offset the positive influences of the warming and wetting climate in the western and central regions; this confirms our previous studies that human activities could mask or aggravate the impacts of climate change (Hao et al., 2016a, Liu et al., 2017). The present study represents a process toward separating the grazing contribution from climate effects at pixel scales. However, the small variation of the vegetation LAI in winter may cause overestimation of grazing intensity and grazing duration.

4.3. Effects of grazing on shrub vegetation

With the implementation of grazing exclusion and grazing rotation policy in grasslands, shrub lands have been widely used for grazing.

Therefore, grazing areas have been extended to the forest areas (Guo et al., 2003), which resulted in sharp declines in the LAI of shrub lands during 1985–1991 and 1997–2004 (Fig. 4). The results also show that shrubs in the eastern area carried the maximum grazing density during 2001–2010 (Fig. 6). Our results are consistent with the observed dramatic vegetation degradation in the main part of the Qilian Mountain Nature Reserve during the 1990s, with dominant degraded vegetation in shrub and grassland (Guo et al., 2003). Wang (2005) reported that the overgrazing in this region had led to the expansion of grazing to shrubbery and sparse woodland areas. Approximately 30% of the shrub lands are considered degraded in the eastern part of the Qilian Mountains.

Shrub lands play a unique role in forest-grassland ecosystems in the Qilian Mountains because of their important role in soil and water conservation and the regulation of river runoff (Wang, 2005). The shrubbery area in the Qilian Mountains is large and widely distributed, 2.3 times that of the spruce forest (Guo et al., 2003; Yang et al., 2008). Therefore, the reduction of shrubbery will have a negative feedback effect on the ecological environment of the Qilian Mountains.

4.4. Regional dynamic grazing detection methodology

The original dynamic reference-cover method (Bastin et al., 2012) provided an effective approach to determine the persistent LAI pixel locations (or baseline pixel locations) with minimal grazing disturbance, without the need for information from reference sites. However, this method does not consider grazing intensity and duration. Wang et al. (2016) developed an approach to estimate regional grazing density and duration using NDVI data; in that method, annual ground-based nongrazing reference pixel locations must be provided when determining the grazing locations. However, for regions with massive nomadic (i.e., no fixed grazing point) and intensive grazing activities (e.g., the Heihe River basin), long-term nongrazing reference point series data are difficult to obtain. Therefore, this method is limited to studying long-term grazing dynamics at the regional level. The dynamic reference-cover method is an efficient solution. Our study integrated the above two methods and developed a pixel-scale dynamic grazing pressure method while taking persistent LAI pixel locations as nongrazing reference points with different vegetation types. We consider this methodology to be an important advance for monitoring vegetation dynamics and making management decisions in grazed ecosystems of semiarid mountain regions.

4.5. Uncertainties

The estimation of LAI from remote sensing data are the only feasible way to generate LAI products at regional scales (Liang et al., 2013). Currently, multiple global LAI products have been generated from various types of satellite remote sensing data and retrieved using various methods and possessing different spatial resolutions from 1 km to 10 km (Ganguly et al., 2008). The GLASS LAI product spans the period 2000–2013 and was derived from MODIS surface reflectance data at a relatively high temporal resolution of eight days and a spatial resolution of 1 km. Studies have shown (Xiao et al., 2016) that the GLASS product is of higher quality and accuracy than the existing products and has much longer time series; it is thus highly suitable for various environmental studies.

Due to the lack of spatially explicit grazing data, we cannot directly validate the results in this study; however, that is the reason why we developed the approach, to determine the baseline pixel locations with minimal grazing disturbance without the need of information from reference sites. Indirectly, we offer circumstantial evidence below for the results that confirm that the method used in this study was carefully considered and appropriate.

First, the existing research supports our results (Guo et al., 2003; Wang, 2005). Second, some field investigations have been carried out at some sites, e.g., Zhangye and Yeniugou, in the upper Heihe river basin, which also support our findings (Su et al., 2010; Ning and He, 2006; Hao et al., 2016a, 2016b). Third, although the 'Gridded Livestock of the World' (GLW) dataset from the FAO (Wint and Robinson, 2007) has a coarse spatial resolution of 0.05°, it provides regional grazing distribution created through the spatial disaggregation of sub-national statistical data (Han et al., 2014) that can be used to indirectly verify our results. The annual grazing density pattern during 2001–2010 in this study is roughly consistent with that in the GLW dataset in 2005 (See S2) in the central and eastern areas of the basin. The reason that the distribution in the west is not well-matched is that the GLW dataset includes small-ruminant data only; data for large ruminants, such as yaks, which are extensively grazing in Yeniugou, are not included.

5. Conclusions

The long-term dynamics of regional grazing stresses and LAI were examined using high-resolution satellite-derived vegetation data. Overgrazing was the dominant driver for the reduction in vegetation LAI for alpine grassland and shrubs in the headwaters of the Heihe River basin. Grazing management (e.g., grazing exclusion and grazing rotation) in recent decades has contributed to the overall improvement of LAI. However, overgrazing has posed additional challenges to the shrub-grassland ecosystem recovery in the eastern part of the study watershed, which has high grazing density as well as unfavorable climate.

Our analysis indicates that there was strong evidence that grazing aggravated the negative effects of warming and drying climate on the vegetation LAI in the eastern region, whereas grazing masked the positive effects of warming and wetting climate in the western and central areas. Without considering the long-term grazing change, the positive contribution of climate could have been underestimated. Therefore, the current grazing pressure should be alleviated, and grazing management needs to be further strengthened in this basin.

In the past decades, the extent of environmental degradation in the vast expanse of grasslands in northwestern China has continuously increased, and overgrazing is the major contributor (Baranova et al., 2016). Our results show that detecting regional grazing effects is needed to estimate the extent and dynamics of overgrazing and to elaborate a scientific basis for developing grazing strategies to avoid overexploitation of grasslands and to develop appropriate ecological solutions to improve the service function of the mountainous grassland ecosystem. From the perspective of pasture management, the current study provides a foundation for future strategy-oriented research. To improve the ecosystem services of the mountainous grassland ecosystems, grazing pressure should be reduced, particularly for the eastern region with aggravated coupled negative effects of unfavorable climate and overgrazing. Rotation grazing should be incorporated into management plans that generally specify a reduced grazing duration within the overall grazing season. In addition, the insights obtained from our study about the specific locations that were most impacted by grazing or climate are useful for evaluating the effectiveness of the recent massive ecological restoration campaigns in this region. Such information may help decision-makers better manage rangelands and improve ecosystem productivity and water supply in arid and semiarid mountainous rangeland. Our findings have broad and important environmental implications for mountainous watersheds in China and other regions with a similar ecosystem, e.g., the Eurasian highlands (Sun and Liu, 2013). Future studies should focus on understanding the biophysical processes that affect LAI dynamics for different ecosystems.

The dynamic grazing pressure method developed from this case study is an efficient approach to quantify long-term grazing activities at a regional level. We consider this methodology to be an important advance in monitoring spatiotemporal grazing distribution dynamics and a contribution to ecosystem service science. It also provides useful information for rangeland managers about the sustainable use of grasslands in mountainous grazed ecosystems in the arid and semiarid regions. The

method should be further evaluated with field data from similar regions to improve its applicability. Similarly, the methods for separating grazing effects on vegetation dynamics from climate effects should be evaluated with process-based ecosystem models in future studies.

Acknowledgments

Conflicts of interest

The authors declare that they have no conflict of interest.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.05.224.

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